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POTENTIAL EFFECTS OF RADIOACTIVE RELEASES TO THE AQUATIC ENVIRONMENT[1]

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ABSTRACT

The increasing amounts of radioactive waste presently being stored and the possibility for its accidental or controlled release to the aquatic environment emphasizes the need for assessing the potential dangers of radioactivity upon fishery organisms. In addition, there is a continuing possibility of radionuclides being released at any point in the fuel cycle from the mining of ore to the reprocessing of fuel elements after use in nuclear power reactors. Once radioactivity enters the environment, only physical decay can eliminate it. Radioactivity released to the aquatic ecosystem is taken up by both the biotic and abiotic components with most becoming associated with the abiotic portion. That portion which is accumulated by the biota is cycled through the food web and may be concentrated at any trophic level since organisms have different requirements or capacities to accumulate the different radionuclides. One of the most important concerns is how much radioactivity is required to affect adversely aquatic organisms. Both short- and long-term exposure, depending upon the level of radioactivity, can affect individual organisms as well as populations. Thus there is need for concern about the future handling of radioactive waste and the use of nuclear power due to their potential for impacting the environment. Decisions on radioactive waste disposal and the use of nuclear power should be based upon rationalism, not emotionalism.

RÉSUMÉ

Les quantités croissantes de déchets radioactifs que l'on entrepose actuellement en milieu aquatique et la possibilité que ces déchets y soient libérés accidentellement ou sous contrôle accentue la nécessité d'évaluer le danger potentiel de radioactivité pour les organismes marins. En outre, il y a risque permanent de rejet de radionuclides à n'importe quel stade du cycle, depuis l'extraction du minerai jusqu'au retraitement des déchets radioactifs. Une fois que la radioactivité a pénétré dans le milieu, seule la dégradation naturelle peut l'éliminer. Un certaine partie de la radioactivité libérée dans l'écosystème aquatique est absorbée par les composants biotiques mais la majeure part s'associe aux éléments abiotiques. La portion qui est accumulée par les organismes vivants parcourt toute la chaîne alimentaire et peut être concentrée à n'importe quel niveau trophique car les organismes n'ont pas les mêmes besoins ni les mêmes capacité d'accumuler les différents radionuclides. Un des points les plus importants à connaître est la dose de radioactivité qui est nuisible pour les organismes aquatiques. L'exposition, tant courte que de longue durée, selon le niveau de radioactivité, peut affecter les divers organismes ainsi que les populations. Ainsi, il faut se préoccuper pour l'avenir de la manutention des déchets radioactifs et de l'utilisation de l'énergie nucléaire étant donné les risques potentiels qu'ils représentent pour l'environnement. Les décisions concernent l'élimination des déchets radioactifs et l'utilisation de l'énergie nucléaire devraient être prises sur des bases rationnelles et non émotives.

INTRODUCTION

In recent years, man has added significant amounts of radioactivity to the environment, which could be potentially damaging to human beings and to the biota in freshwater and the oceans. Since many aquatic organisms are used as food by humans, and these organisms can accumulate large amounts of radioactive materials relative to their concentrations in the water, there is need to know the rates and levels of accumulation of radionuclides by the aquatic organisms. Further, since there is a possibility that radioactivity can be released in large quantities in limited areas of the aquatic environment

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as a result of accidents, there also is a need to know the amounts of radionuclides that can be returned in fishery products and the levels of radioactivity that may result in significant radiation effects on aquatic organisms.

The aquatic environment represents more than three-fourths of the earth's surface and has received a proportionate amount of fallout from nuclear weapons testing. Much of the fallout on land has leached from the soil, and has been carried through runoff to freshwater streams, and then to the oceans, while other radioactive materials have been released directly into the aquatic environment from nuclear reactors and radioactive waste outfalls. The oceans, which serve as a sink for many pollutants, may be destined to receive more radioactive wastes now stored on land. At the present time, the U.S. Government is considering the possible use of the seabed at great depths as a place to dispose of high-level radioactive wastes in specially designed containers.

Radioactivity is of concern because our senses cannot detect it and because radiation effects on man are being observed as a result of the atmospheric explosion of nuclear weapons during and following World War II. Also, once radionuclides are released into the aquatic environment, only the passage of time can reduce their radioactivity. Each radionuclide has its own rate of decay or half-life (time required for one half of that present to decay), which ranges from minutes to thousands of years. The detection of radioactivity in fish (Saiki et al. 1957) first made man aware that contaminants released into the oceans could return to him in seafood. Prior to this, the oceans had been considered so large that any unwanted material could be disposed of there safely. Because of the suspected dangers to man from such disposal, funding in the U.S. became available from the federal government for research on the cycling and effects of radiation so that radionuclides became the best understood and controlled of all the pollutants that man has produced.

The accumulation of radionuclides and their transfer through food chains have been followed in both laboratory and field studies (Cross et al. 1975). In the laboratory, it has been possible to determine the effects of food, feeding, temperature, salinity, pH, isotope dilution, and concentration of chemically similar elements upon the accumulation, assimilation and tissue distribution of radionuclides. Also, accumulation and food chain studies have been carried out in the aquatic environment where radionuclides have been released through nuclear weapons testing, reactor operation and radioactive waste release.

The effects of radiation upon aquatic organisms and upon humans from eating these organisms also has been intensively investigated (Foster et al. 1971; Templeton et al. 1971). Laboratory experiments at relatively high levels of radioactivity and field observations in areas where radioactive wastes have been discharged over long periods of time have been used in an effort to determine doses of radiation that are harmful to organisms. The restrictions that have been set by several nations on levels and rates of release of each radionuclide to the aquatic environment are based upon the radionuclides return to man from drinking water and food.

While this paper is concerned with the cycling of radionuclides and effects of radiation, it should be emphasized that it is not an exhaustive review but covers primarily those aspects of radioecology of relevance to fishery biologists who are not familiar with this discipline.

SOURCES OF MAN-MADE RADIONUCLIDES

There are a number of sources of man-made radionuclides (Table 1) that are derived either directly from nuclear fission or fusion, or indirectly through neutron bombardment of stable elements (induced) (Table 2). The largest amount entering the aquatic environment to date, by more than two orders of magnitude, has been from atmospheric fallout from weapons testing (Preston 1972). Probably the greatest future source of radioactivity to the aquatic environment will be from nuclear reactors, assuming that the world will not become involved in a nuclear war and that atmospheric nuclear testing does not resume to any significant level. Because of the volumes of water required to cool nuclear reactors, most have been built on lakes, rivers, estuaries and oceans. The majority have been designed to produce electrical power, nuclear fuel and explosives for weapons. The type and size of the reactor will determine the amount and type of waste that may be released into the aquatic environment (Blaylock and Witherspoon 1978). Releases of radionuclides to the environment from a power reactor originate primarily from scheduled low-level emissions during normal operation, but there is always the possibility of accidental releases. With the exception of accidents, the annual releases of radioactivity have accounted for only small per-

Table 1. Sources of radionuclides to the aquatic environment through man's activities (modified from Joseph et al. 1971).

Source	Activity
Nuclear fuel cycle	Mining, uranium processing, fabrication, spent fuel reprocessing
Nuclear reactors	Electric power generation, ships, satellite power, research, plutonium production, wastes
Nuclear explosives	Military and civilian applications
Encapsulated radioisotopes (power)	Marine navigation aids, weather stations, artificial human organs
Encapsulated radioisotopes (radiation)	Medical radiology, industrial radiography, research
Radionuclides	Medical uses and research activities

Table 2. Important man-made radionuclide in the aquatic environment and their half-lives (modified from Woodhead 1973).[a]

Hydrogen-3	12,3y
Carbon-14	5 730y
Manganese-54	303d
Iron-55	2,6y
Cobalt-57	270d
Cobalt-60	5,26y
Nickel-63	92y
Zinc-65	245d
Strontium-89	52d
Strontium-90	28,1y
Zirconium-95	65,5d
Niobium-95	35,0d
Ruthenium-103	39,5d
Ruthenium-106	368d
Silver-110 m	255d
Antimony-125	2,7y
Iodine-131	8,05d
Cesium-137	30,0y
Cerium-141	33d
Cerium-144	284d
Promethium-147	2,62y
Europium-155	1,81y
Plutonium-239	23 309y

[a] d = day, y = year

centages of the legal permissible concentrations (Rice and Baptist 1974).

Radioactive wastes can be disposed of as low-level liquid wastes or as high-level encapsulated wastes. Liquid wastes can be released on a routine basis from reactors or nuclear fuel reprocessing plants. These releases should fall within the International Commission on Radiological Protection (ICRP) guidelines for releases of radioactivity. Some of the most abundant liquid wastes may contain tritium (hydrogen-3), the fission products strontium-90, zirconium-niobium-95, cesium-137, cerium-144 and the induced radionuclides manganese-54, iron-55, cobalt-60 and zinc-65. High-level packaged wastes have been disposed of on the deep ocean seabed. Included in these wastes are radionuclides that have half-lives on the order of tens of thousands of years. Consideration is now being given to burying these wastes within the deep seabed sediments, which will reduce the probability of their release to the water column and their return to surface waters.

CYCLING

The biogeochemical processes that control the distribution of stable elements in the aquatic environment also control the distribution of radionuclides. Upon entering the aquatic environment, radionuclides can remain in solution or in suspension, precipitate and settle to the bottom, or be taken up by plants and animals. Certain processes interact to dilute and disperse these materials, while other processes simultaneously tend to concentrate them. Currents, turbulent diffusion, isotopic dilution and biological transport dilute and disperse radionuclides. Concentrating processes may be biological, chemical or physical. Radionuclides are concentrated biologically through uptake and assimilation by aquatic organisms and chemically and physically by adsorption, ion exchange, co-precipitation, flocculation, and sedimentation through the interaction of such biotic and abiotic processes. Radionuclides also are cycled through water, sediment, and biota, and each radionuclide tends to take a characteristic route and rate of movement through these components or reservoirs of the aquatic environment (Rice et al. 1965; Wolfe et al. 1973).

In freshwater systems and estuaries, chemical and biological processes tend to concentrate radionuclides. At the freshwater-saltwater boundary zone in estuaries, hydrological and physiochemical conditions can significantly influence the availability of radionuclides to the biota (Cross and Sunda 1978). The shallowness of estuarine and most freshwater habitats enhance the role of benthic communities in the exchange of radionuclides between sediments and water (Wolfe and Rice 1972). Such factors must be considered in estuaries, coastal waters and freshwater environments to a greater extent than in the open ocean. In the open ocean and deep lakes, however, thermal stratification, depth of water and circulation patterns make the sediments and benthic communities relatively unimportant in the cycling of radionuclides. In these systems, elements that are exchanged between the sediment and water may take thousands of years to reach the surface waters.

Sediments may accumulate radionuclides through the physical processes of exchange and adsorption (Duursma and Gross 1971). In effect, sediments and biota compete for radionuclides present in water. Although in some instances, sediments initially remove large quantities of radionuclides from the water and thus prevent their immediate uptake by the biota, this sediment-associated radioactivity later may affect many benthic organisms by exposing them to radiation (Woodhead 1973). Radionuclides also leach from the sediments back to the water and again become available for uptake by the biota. Even though radionuclides are associated with the sediment they may become available to the biota due to variation in the strength of the binding between the different radionuclides and the sediment particles. Loosely bound radionuclides on sediments can be stripped from particles of sediment and utilized by bottom-feeding organisms ingesting sediments (Luoma and Jenne 1976).

Aquatic plants and animals also play an integral part in the cycling of radionuclides (Fig. 1). They accumulate radionuclides by adsorption, absorption and ingestion. Conversely, radionuclides can be lost by desorption, excretion and decomposition. For example, even if an organism accumulates and retains radionuclides but dies, the radionuclides will be released back into the environment through organisms that decompose the dead organic material into its elemental components (Rice and Baptist 1974). In addition, radionuclides that are adsorbed or ingested but

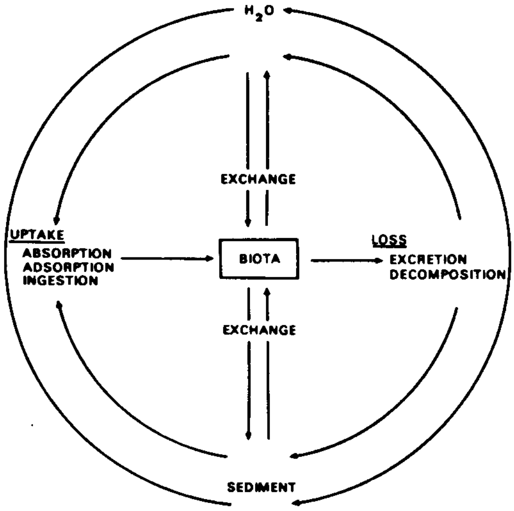

Fig. 1. Processes involved in uptake and loss of radionuclides by marine biota (after Rice and Baptist 1974).

not assimilated by aquatic animals can be transported downward in the water column with fecal material (Osterberg et al. 1963) or in cast exoskeletons of pelagic crustacea (Fowler and Small 1967).

The extent to which fish can accumulate radionuclides depends upon their availability and the physical state of the radionuclides in the water. Radionuclides from food have been shown to be more available to fish than from the water (Jeffries and Hewett 1971; Pentreath 1973a, 1973b, 1973c). Several fission products—ruthenium-106, cerium-144 and zirconium-niobium-95—which are relatively insoluble, are poorly absorbed across the gut wall of fish (Pentreath 1973d). Baptist and Hoss (1965) found that less than 1% of the cerium-praseodymium-144 ingested by croaker *Micropogon undulatus* was assimilated. The biologically significant induced radionuclides—manganese-54, iron-55, cobalt-60 and zinc-65—are assimilated much more readily across the gut wall than are the fission products discussed above (Osterberg et al. 1964; Cross et al. 1975). In addition, freshwater fish also have the capacity to accumulate relatively high levels of the radionuclides of strontium and cesium due to the relatively low levels of calcium and potassium in fresh water (Preston 1972). Cross et al. (1975) have reviewed the transfer of radionuclides through marine food webs leading to fish.

EFFECTS OF RADIOACTIVITY

While the accumulation of radioactivity by aquatic organisms is of importance because of their use as food by man, concern also has been expressed about the exposure of aquatic organisms to radioactivity from the water, sediments and from the radioactivity contained within the organisms themselves. To predict the possible effects of accidental releases of radioactivity on individual organisms and populations, it is necessary first to know if releases to date have had any impacts. Various approaches to the study of the effects of ionizing radiations on aquatic organisms have been used. For example, acute and chronic laboratory exposures, the study of animal populations in areas receiving radioactive waste and population dynamic models utilizing stressed populations (Ophel et al. 1976; Templeton et al. 1976; Blaylock and Trabalka 1978) have been used in an effort to reach a better understanding of radiation levels harmful to aquatic organisms.

In recent years, several review articles have summarized laboratory radiation experiments on aquatic organisms where acute radiation doses were used, and both lethal and nonlethal responses were examined (Polikarpov 1966; Rice and Wolfe 1971; Templeton et al. 1971; Chipman 1972; Rice and Baptist 1974; Ophel et al. 1976; Blaylock and Trabalka 1978). As Ophel et al. (1976) pointed out, most laboratory experiments involve acute exposures to radiation delivered over a short period of time. The data generated by such experiments show that aquatic organisms have LD_{50}s (50% lethal dose), which range from 10^2 to 10^6 rads,[2] and relate very poorly to environmental situations (Cross 1978; Woodhead et al. 1976) where the dose rates are low (0,6 to 3 400 micro rads/h) and the time periods are long (> year).

Attempts have been made to design chronic laboratory irradiation experiments so that exposures include a significant portion of the organism's life cycle. These experiments have used both sealed external cobalt-60 and cesium-137 sources and radionuclides released into the water (Blaylock and Trabalka 1978). Donaldson and Bonham (1964, 1970) irradiated coho and chinook salmon eggs (*Oncorhynchus kisutch* and *O. tshawytscha*) from fertilization through 80 days and looked for differences between the irradiated and control groups. No demonstrable effects were noted at a dose rate of 0,5 R/day,[3] but at 10 R/day significant changes in sex ratios were observed, and at 20 R/day no females occurred. Other investigators have used fish, crabs, and snails in laboratory chronic irradiation experiments and have shown few adverse effects except at the highest dose rates used (Engel 1967; Cooley and Miller 1971; Kaufman and Beyers 1973).

In other attempts to design irradiation experiments that can be related to environmental radioactive contamination, some investigators have exposed different developmental stages of aquatic organisms to radionuclides dissolved in seawater. Developing eggs of several species of fish have been used extensively because eggs have been shown to be the most sensitive stage in the life cycle of fish. For example, Brown and Templeton (1964) did not observe any significant effect of protracted irradiation on the hatching of eggs or development of the plaice *Pleuronectes platessa* at dose rates ranging from 0,01 to 1 R/h or to total dose up to 500 R.

Some investigators have attempted to produce radiation damage in fish through the use of high concentrations of internally deposited radionuclides. For example, rainbow trout *Salmo gairdneri* showed no detrimental effects from body burdens of zinc-65 and phosphorus-32 from 100 to 10 000 times greater than the concentrations occurring in the Columbia River in 1965 (Foster and Soldat 1966). At higher levels, however, both zinc-65 and phosphorus-32 caused damage to the blood cell producing tissues in the trout (Nakatani 1966). A discussion of effects from internally deposited radionuclides and external radiation on aquatic organisms has been compiled by Templeton et al. (1971) and Ophel et al. (1976).

One of the most direct methods of obtaining an understanding of the impact of radioactive releases on natural populations is to study, intensively, environments where radioactivity has been released intentionally. By following changes in population size of resident organisms in such areas, it should be possible to determine the biological effects of accidental releases. Three such areas where intentional releases have been studied in considerable detail are the

[2] Rad = absorbed dose of ionizing radiation from alpha, beta, and gamma rays.

[3] R = roentgen, a unit of exposure dose for gamma rays or X-rays.

White Oak Lake at the Oak Ridge National Laboratory, Tennessee, the Columbia River downstream from the Hanford weapons production plant in Richland, Washington, and the Irish Sea in the vicinity of the Windscale reprocessing plant.

White Oak Lake was established as a radioactive waste settling basin for the production reactors at Oak Ridge, and as a result the organisms in the lake have been exposed to long-term chronic irradiation from 1943 to the present. Numerous investigations have been conducted on the fish and invertebrate populations of the lake. These investigations primarily have been concerned with the effects of ionizing radiations on the gene pools of aquatic populations.

Three natural populations of aquatic organisms studied intensively from 1963 to the present are the midge *Chironomus tentans*, the snail *Physa heterostropha*, and the mosquitofish *Gambusia affinis*. These species were exposed to low-level chronic irradiation for many generations. The *Chironomus* population was studied from 1960 through 1970 and was exposed to decreasing dose rates that ranged from 230 rad/year to 11 rad/year (Blaylock and Trabalka 1978). The irradiation caused an increase in the frequency of aberrations in the giant salivary chromosomes of the midge larvae and the frequency of aberrations was dose rate dependent (Blaylock 1966). When the dose rate decreased to 11 rad/year or less due to the decay of the radioactivity in the lake, the numbers of aberrations decreased and did not differ from the control population. Cooley and Nelson (1970) and Cooley (1973), using a natural population of the snail *Physa heterostropha*, demonstrated that a dose rate of 0,65 rad/day reduced egg capsule production when compared to an unirradiated group. Egg production, however, was similar in both groups because there were more eggs per capsule in the irradiated population. The genetic variability of the mosquitofish *Gambusia affinis* also has been studied intensively (Blaylock 1969; Blaylock and Frank, in press). When irradiated and controlled populations of fish were compared, a larger brood size occurred in the irradiated population; there also was a significantly higher frequency of dead and abnormally formed embryos. It was suggested that the increased fecundity of the irradiated fish population compensated for the increased mortality caused by radiation. Trabalka and Allen (1977), however, demonstrated that laboratory-reared fish from the irradiated and controlled populations had the same fecundity but that the irradiated fish had many deleterious genes. At the same time the population in the field was thriving, which indicated that measurements of radiation damage such as genetic load may not be a valid index of population fitness.

The construction of the Hanford plutonium production reactors on the Columbia River resulted in the average release of 1 000 Ci/day[4] of radioactivity directly into the River (Osterberg 1975). This plant used river water for once-through cooling, and the radionuclides released were primarily from the neutron activation of stable elements in river water, antifouling agents and corrosion products. The isotopes of greatest importance were phosphorous-32, chromium-51 and zinc-65. From the time the plant began operation in the mid-1940's until the last production reactor was shut down in 1971, relatively large quantities of radioactivity were released into the river. Large-scale research projects at universities and neighboring government institutions were organized to follow the cycling of the released radioactivity in the river, estuary and adjacent Pacific Ocean. The overall result was that no effects on the aquatic biota were detected from the radioactivity accumulated by various organisms of the food chains in the river, estuary or adjacent ocean (Osterberg 1975). Much valuable information on the partitioning of radionuclides in aquatic ecosystems and the coastal oceanographic processes of the river plume, however, came from these investigations.

Since about 1956 the Windscale reprocessing plant and reactors on the west coast of England have released low-level radioactive waste into the northeast Irish Sea, a productive fishing ground. During investigations conducted to determine whether released radioactivity would affect the resident population of plaice *Pleuronectes platessa*, Woodhead (1970) calculated the doses of radiation a developing plaice egg would receive from the sediment and from radioactivity adsorbed to the chorionic membrane. Laboratory experiments were conducted at ambient and higher dose levels to determine the effects on hatching and larval development. No significant damage occurred at a total integrated dose

[4] Ci = Curie, the basic unit measurement for radioactivity, equal to $3{,}7 \times 10^{10}$ atomic disintegrations per second.

to the eggs of approximately 0,18 rad. In the environment the total dose rate to the eggs from released radionuclides was $9{,}32 \times 10^{-2}$ micro rad/h, while the dose from the natural occurring radionuclide ^{40}K was $6{,}96 \times 10^{-1}$ micro rad/h is almost an order of magnitude greater. Templeton et al. (1976) concluded that the release of radioactivity to the environment at Windscale had not had any measurable effect upon the plaice population. These investigations have demonstrated that at higher levels of radioactivity in the natural environment, it was difficult to demonstrate any effects on the resident populations of organisms. Although no damage to population structure was detected, however, it does not eliminate the possibility that subtle changes in an individual's genome might have occurred. In nature, probably almost any mutation that weakens an animal will cause its death and therefore will result in the mutation being eliminated from the gene pool.

The failure to detect radiation-induced somatic changes at either individual or population levels in contaminated environments does not necessarily prove that no effects have occurred. Instead, it may reflect our lack of understanding of the natural variability within the system. It can be stated, however, that no catastrophic mortalities have occurred and that the subtle changes, which might be suspected, were not observed due to long-term fluctuations in the ecosystem.

In attempts to use fishery statistics and data, Zaystev and Polikarpov (Rice and Angelovic 1969) calculated the length of time required to reduce fish populations 50% if various percentages of the spawned eggs were killed by radiation. The calculations assumed constant levels of mortalities (i.e., 10% a year), constant levels of fishing and a set recruitment rate dependent upon spawning population size. Such assumptions are not valid, however, for many populations of marine fish because of natural and man-induced variability (Beverton and Holt 1957).

Possible effects of radiation at the population level were investigated by using knowledge of the population dynamics of selected commercial fish species by an International Atomic Energy Agency Panel in a report entitled "Effects of Ionizing Radiation on Aquatic Organisms and Ecosystems" (Templeton et al. 1976). The panel considered the role of density-dependent mortality in the stock-recruitment relationship in marine populations of both high and low fecundity (Fig. 2). Using commercially exploited fish stocks as an example, the authors concluded, "If mortality of eggs is being enhanced by the low levels of irradiation presently existing in the marine environment, then recruitment to the stocks of highly fecund marine species of fish is unlikely to be adversely affected unless those stocks are already at risk because of severe over-exploitation." In other words, the result of mortalities of eggs caused by irradiation would decrease larval competition for food and space and, therefore, would increase the probability of survival for the remaining individuals.

Survival rates for highly fecund density-dependent fish stocks increase dramatically at low stock sizes (Fig. 3). The curve demonstrates the relationship between spawning stock size and survival for Atlantic menhaden *Brevoortia tyrannus* for the 1955–1970 year classes. Each point on the graph shows by year the estimated egg production and percent survival to age one. Similar population responses could occur, however, if some perturbation such as radiation was causing mortalities of eggs and larvae. This would ultimately reduce spawning stock size. Obviously, we would not expect the density-dependent relationship to compensate for radiation-induced mortalities in severely exploited fish stocks.

Evidence also exists that increased pressure on fish stocks by over-exploitation or other stresses may be compensated for by an increase in fecundity of surviving adults. Blaylock (1969), for example, reported that the mosquitofish *Gambusia affinis* increased fecundity relative to

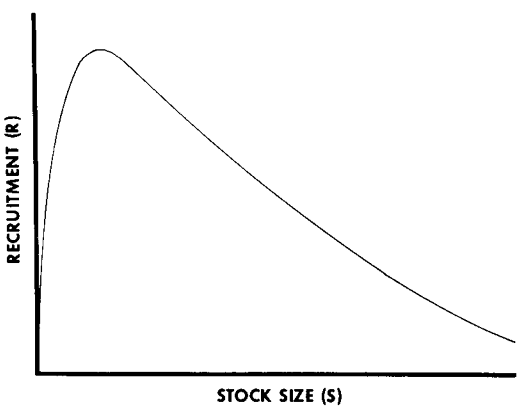

Fig. 2. An extreme example of a density-dependent relationship between spawning stock size and recruitment in highly fecund species (figure from Beverton and Holt 1957).

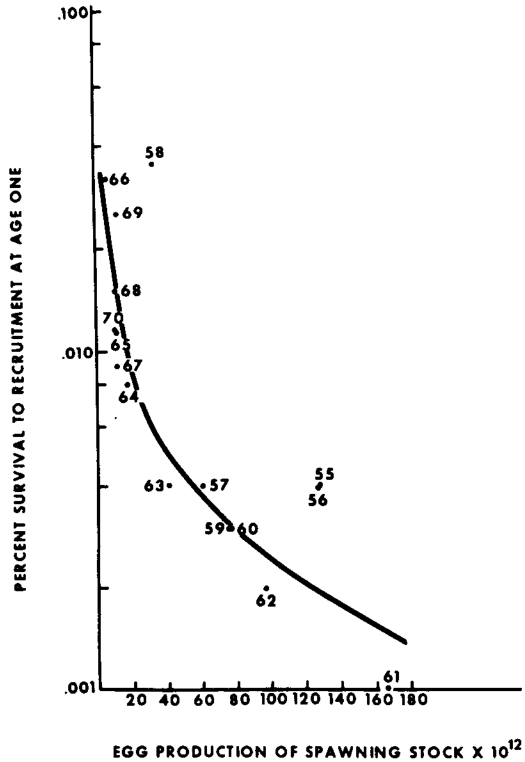

Fig. 3. Relationship between spawning stock size and survival for Atlantic menhaden, *Brevoortia tyrannus*, for 1955–1970 year classes (W. Nelson, personal communication).

controls in the presence of chronic exposure to radioactivity in a freshwater environment.

The inherent dynamics of highly fecund marine populations, therefore, could compensate for additional mortalities of young caused by contaminants such as radionuclides. This compensating mechanism is limited in its capacity to "protect" a species that experiences high mortalities in its early stages. The actual level of mortality that must occur to affect a population significantly will be highly variable and dependent on a number of additional factors such as predation, food supply, exploitation, etc.

EXPOSURE OF MAN TO RADIOACTIVITY

Since it generally is accepted that any radiation dose above background levels to man will be accompanied by some risk of deleterious effects, releases of radioactivity to the aquatic environment must be kept to a minimum. To ensure that man's exposure to radiation will be minimal, restrictions have been placed upon release and disposal of radioactivity into the aquatic environment. These restrictions have been established through the efforts of the International Atomic Energy Agency and the International Commission on Radiation Protection.

In both the U.K. and the U.S., two different approaches have been taken to control routine releases of radioactivity into the aquatic environment. In the U.K. the critical pathway approach is used (Preston 1969), and in the U.S. the concept of maximum permissible concentration (MPC) is used (Code of Federal Regulations 1967). In the critical pathways approach, the pathways by which radionuclides can reach man are first identified and evaluated in relation to the level of radionuclides that can be transferred to man. The dose of radiation to man then is calculated for the "critical group" exposed. Radiation doses from any other pathway for that radionuclide cannot exceed those levels. An excellent example is consumption of ruthenium-106 contaminated seaweed *Porphyra* in laverbread by a small coastal population near Windscale. This pathway sets the limits on ruthenium-106 discharges. Other examples of critical pathways to man in the U.K. are shown in Table 3. In the U.S. the concept of MPC is set forth in the "Code of Federal Regulations," Title 10, Part 20. This limits, on a daily basis, the concentration of radionuclides in 2,2 liters of drinking water to that amount (contained in 2,2 liters of water per day), which will not exceed the maximum permissible radiation dose (ICRP 1959 and NCRP 1959).

Another approach is the use of specific activity for establishing permissible levels of environmental radioactivity. This was first recommended for radioactive waste disposal into the U.S. Pacific coastal waters (NAS-NRC 1962). The application of this approach is based mainly on two assumptions: (1) that a radionuclide introduced into the environment readily equilibrates with the non-radioactive element such that biological concentrating mechanisms will be unable to discriminate between different forms of the element, and (2) that the quantity of each stable element in each body organ is constant and does not fluctuate with intake of that element. Thus, the distribution of radionuclides will correspond to the distribution of the stable element.

In the calculations for radiation dose limits to

Table 3. Critical organisms and radionuclides in the pathways to man at four radioactive waste discharge locations in the U.K. (modified from Preston and Mitchell 1973).

Site	Critical organisms	Critical radionuclide
Windscale	*Porphyra* (seaweed)	Ruthenium-106
	Fish	Cesium-137
		Cesium-134
Bradwell	Oyster	Silver-110
		Zinc-65
Dungeness	Fish	Cesium-137
		Cesium-134
Hinkley Point	Fish & shrimp	Cesium-137
		Cesium-134

man from incorporated radionuclides, the ICRP (1959) used the "standard man." This model was based on a middle-European man with food habits of that geographical area. Tanaka et al. (1979) pointed out that such standards were not applicable to the Japanese because of differences in both body structure and culture. The primary difference is that the Japanese standard man has very different patterns of food consumption. Since "he" consumes many times more fishery products and seaweed than a European, Tanaka suggested that the standards should be designed to take into account the peoples and customs of the various cultures around the world. For example, a certain level of release of radioactivity may not affect a beef and potatoes food culture but could seriously impact a culture where fishery products are the primary protein source.

CONCLUSIONS

The sudden and awesome beginning of the nuclear age during World War II and subsequent atmospheric nuclear testing gave rise to intensive international research on the environmental behavior of radionuclides and biological damage from radiation. Results of this research have been partially responsible for the establishment of extremely tight restrictions governing all uses and releases of radionuclides into the environment. As a result, peaceful uses of radioactivity have not interfered with commercial and recreational harvest of fisheries resources in either the freshwater or marine environment.

This has not been the case, however, with other pollutants. Excessive discharges of mercury and cadmium, for example, have resulted in adverse effects on man (Douglas-Wilson 1972; Kurland 1973). Discharges of these heavy metals have caused the closing of a number of rivers and lakes to recreational fishing in the U.S., Canada and Sweden. Even several years after environmental discharges of mercury had been sharply reduced or eliminated, certain species of recreational fish from several freshwater areas of the U.S., e.g., Holston River, Shenandoah River and Lake St. Clair, still cannot be eaten. In addition, excessive releases or uses of chlorinated hydrocarbons (e.g., DDT, PCB) have impacted the use of freshwater resources, such as salmon in Lake Michigan, and have been linked to the decline of two populations of birds—peregrine falcons and ospreys—along the east coast of the United States (Hickey 1969; Spitzer et al. 1978).

Since the aquatic environment, particularly the ocean, is the final repository for many of man's pollutants and also serves as one of the major sources of protein for the earth's growing population, extreme care must be exercised in the release and disposal of radioactivity. Constant vigil and continued research on the fate and effects of environmental releases of radioactivity must be continued to ensure that misuse of this potentially dangerous contaminant does not occur. In particular, research should be concentrated on obtaining a better understanding of the consequences of elevated levels of plutonium in the aquatic environment (National Academy of Sciences 1975) and developing techniques for damage assessment to fishery resources in the event of a catastrophic release of radioactivity from a nuclear reactor. At the present time, however, there are no known instances where radioactivity has had a deleterious effect on aquatic populations (Templeton et al. 1976; Cross 1978). With the growing number of nuclear power generating facilities and the accompanying wastes, such situations could occur if we do not maintain our vigilence.

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